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## BIOCHARS IN SOILS: TOWARDS THE REQUIRED LEVEL OF SCIENTIFIC UNDERSTANDING

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**Abstract.** Key priorities in biochar research for future guidance of sustainable policy development have been identified by expert assessment within the COST Action TD1107. The current level of scientific understanding (LOSU) regarding the consequences of biochar application to soil were explored. Five broad thematic areas of biochar research were addressed: soil biodiversity and ecotoxicology, soil organic matter and greenhouse gas (GHG) emissions, soil physical properties, nutrient cycles and crop production, and soil remediation. The highest future research priorities regarding biochar's effects in soils were: functional redundancy within soil microbial communities, bioavailability of biochar's contaminants to soil biota, soil organic matter stability, GHG emissions, soil formation, soil hydrology, nutrient cycling due to microbial priming as well as altered rhizosphere ecology, and soil pH buffering capacity. Methodological and other constraints to achieve the required LOSU are discussed and options for efficient progress of biochar research and sustainable application to soil are presented.

**Keywords:** biochar, biodiversity, ecosystem services, ecotoxicology, greenhouse gases, nutrient cycles, policy support, soil organic matter, soil physical properties, soil remediation.

## Introduction

Biochar research has evolved rapidly in terms of published peer-reviewed papers (Verheijen *et al.* 2014), developing a growing body of knowledge to support sustainable decision making (e.g. Lehmann, Joseph 2015). However, many issues have been explored only superficially and there are still considerable knowledge gaps in some areas. For instance, the effects of biochar additions have received more attention than underlying mechanisms; trade-offs between specific mechanisms have only started to be investigated (Jeffery *et al.* 2015a), while long-term interactions in soil ecosystems required to inform sustainable policy development have rarely been addressed (Zhang *et al.* 2016). Many authors reported biochar's potential impacts on agronomic responses or singled out certain environmental aspects but did not consider the wider environmental impacts of biochar use as a soil amendment, e.g. on soil remediation options, soil organic carbon (SOC) stocks, greenhouse gas (GHG) emissions, nutrient leaching, and soil functional diversity, including its use in (agro)forestry.

The current study aimed to identify knowledge gaps and prioritize the focus for future biochar research to inform decision-makers relevant to the full scope of biochar-soil-crop-environment interactions. To achieve this

goal, 36 invited international biochar and soil scientists (Table 1) met over two days in June 2014 at the University of Aveiro (Portugal), to review, discuss and evaluate the current and required Level of Scientific Understanding (LOSU) to achieve sustainable biochar policy development. The workshop was followed by online discussions, leading to an assessment of research priorities based on perceived gaps in scientific knowledge and their importance to different issues relevant to biochar soil amendment.

## 1. Methods

In a preliminary exercise, a methodology using a soil-based ecosystem services approach was proposed, discussed and adopted as a starting point (Jeffery *et al.* 2010, following MEA 2005) to ensure a collective understanding from a policy development perspective and facilitate knowledge transfer of outcomes to a wider audience. Various iterations of plenary discussions and ranking exercises were carried out. Five main thematic areas of biochar research were addressed, which combined the working groups 2 (land use implementation) and 4 (environmental impact assessment) of the COST Action TD1107 "Biochar as option for sustainable resource management"

Table 1. Organisation of the five thematic groups (the names of facilitators/discussion leaders of each thematic group underlined), largely based on the structure of working groups 2 (land use implementation) and 4 (environmental impact assessment) of the COST Action TD1107

Soil biodiversity & ecotoxicology	Soil organic matter & greenhouse gases	Soil physical properties	Nutrient cycles & crop production	Soil remediation
Ana Catarina Bastos	Juergen Kern	Priit Tammeorg	Ellen Graber	Frédéric Rees
António Amaro	Gemini Delle Vedove	Gabriel Gascó	Claudia M. d. S. Cordovil	Georg Lemmer
Alice Budai	Jan Horak	Bruno Glaser	Bruno Glaser	Aline Puga
Xavier Domene	Pietro Panzacchi	Mark Kibblewhite	Claudia Kammann	Flávio Silva
Ciro Gardi	Munoo Prasad	Elena Kondrlova	David Laird	
Simon Jeffery	Verena Lehr	Martinho Martins	Lidia Sas-Paszt	
Susana Loureiro	Giustino Tonon	Wenceslau Teixeira	Greet Ruysschaert	
Marija Prodana		Frank Verheijen	Maurizio Ventura	
			Costanza Zavalloni	

(Verheijen *et al.*, this issue): (i) soil biodiversity and ecotoxicology, (ii) soil organic matter and greenhouse gas (GHG) emissions, (iii) soil physical properties, (iv) nutrient cycles and crop production, and (v) soil remediation. Twenty-two potential soil-based ecosystem services relevant to biochar application as a cross-cutting soil and environmental management tool were identified.

For each of the 22 soil-based ecosystem services (MEA 2005), participants assigned a point score reflecting their assessment of the importance of the service to sustainable policy development and the current and minimally required LOSU of biochar soil amendment interactions with ecosystem services. Subsequently, the main exercise was conducted, using the same structure in the five thematic groups. Here, soil indicators (properties and processes) relevant to biochar were used, instead of soil-based ecosystem services, on account of more consistent understanding between participants from diverse disciplines. Within each thematic group, soil indicators

were identified and ranked according to their relative importance. The relative importance is defined as the extent to which biochar soil amendment affects each indicator. The current and the minimal LOSU considered required to effectively guide policy development were objectively assessed. Key priority issues, specific knowledge gaps and methodological aspects linked to biochar impact evaluation were identified and recommendations for future research were put forward.

The results were refined within each thematic group and the combined data were evaluated according to the following calculations:

- (1) Research Gap = Required LOSU – Current LOSU;
- (2) Research Priority Index (RPI) =  

$$\frac{(\text{Research Gap}) \times (\text{Research Gap}) \times (\text{Relative Importance})}{\text{Research Gap}}$$

Higher values of RPI refer to higher priorities. As the between-group variation in all the above

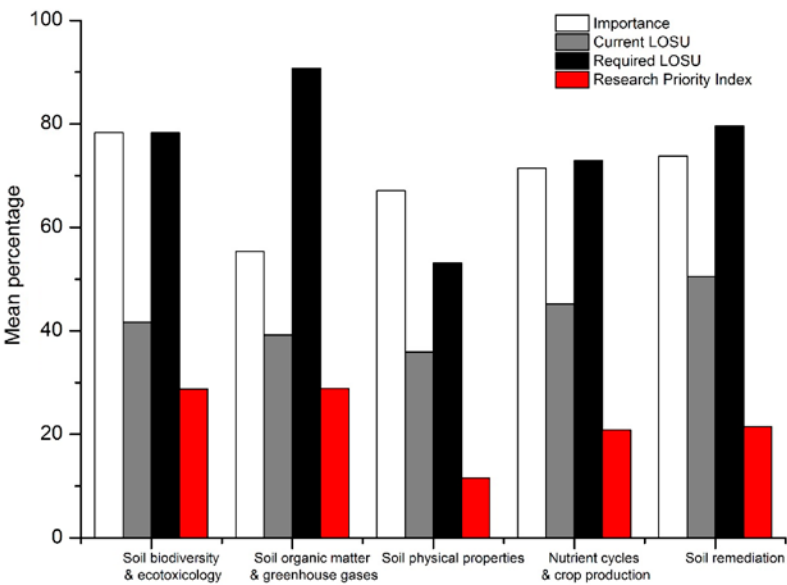


Fig. 1. Differences between thematic groups in the mean values for research priority index and its partitions: Importance, current and required level of scientific understanding (LOSU), and research priority index of individual thematic groups

mentioned values was rather high (Fig. 1), the RPI's of all groups were normalized so that the highest RPI value in each TG corresponded to 100%. Each thematic group was organized through discussion and general consent. A critical number of active participants were required in each group, incorporating various levels of expertise. Each thematic group leader was advised to carefully manage the balance of participants' contributions.

## 2. Results and discussion

### 2.1. Soil biodiversity and ecotoxicology

The highest RPI was identified for biochar capacity to shift a soil community's functional redundancy, which is defined here as the degree of functional overlapping in a given ecosystem. High RPIs were also obtained for assessing the bioavailability of biochar contaminants, trophic interactions, disease suppression, and population dynamics (Fig. 2).

Recent studies focusing on soil-biochar-biota interactions in both terrestrial and aquatic ecosystems suggest potential impacts regarding the application of some biochars to soil on the activity and structure of edaphic and freshwater biological communities (due to leaching and potential transport of biochar particles into watercourse), their trophic relationships, and the processes they mediate. The extent and pattern of biological responses and/or observed toxicity can be assigned to bioavailable biochar fractions and appear to depend mainly on the target species and exposure scenarios, as well as on biochar physico-chemical properties and application rates (Busch *et al.* 2012; Oleszczuk *et al.* 2013; Smith *et al.* 2012; Bastos *et al.* 2014; Marks *et al.* 2014; Domene *et al.* 2015; Jaiswal *et al.* 2015). A number of different mechanisms may play a role in determining biological responses to biochar amendment. These include the potential provision of refuge for microbial communities, shifts in available nutrients and nutrient ratios (Gundale, DeLuca 2006; Prendergast-Miller *et al.* 2014), bioavailable contaminants (e.g. Elad *et al.* 2012; Graber, Elad 2013; Denyes *et al.* 2012; Hilber *et al.*, this issue), enhanced plant root development and systemic defence against biotic or abiotic stress (Elad *et al.* 2010; Jaiswal *et al.* 2014) as well as microbial or plant-symbiont molecular signalling dynamics (Spokas *et al.* 2010; Masiello *et al.* 2013; Graber *et al.* 2015).

Direct or indirect impacts on soil and aquatic fauna have also been reported, which include possible risks to their survival and/or reproduction, trophic relationships and possibly, functional diversity (Ezawa *et al.* 2002; Bastos *et al.* 2014; Domene *et al.* 2014; Thies *et al.* 2015; Marks *et al.* 2016). More work is needed in order to identify specific biochar characteristics and application rates that allow more suitable management and minimization of possible trade-offs between desired benefits and the

short- to long-term integrity of ecosystem functions (Verheijen *et al.* 2014; Hilber *et al.*, this issue).

Much of what is currently known with respect to biochar-biota interactions relies heavily on laboratory-based and/or microcosm approaches, most of which are short-term (Jeffery *et al.* 2015a). Studies focusing on biochar ecotoxicology are often based on acute exposure to high levels of freshly produced, non-modified biochars. Some of such current methodologies poorly represent the natural environment and use individual species testing, such as standardized (e.g., ISO, OECD) bioassays using plants and terrestrial or aquatic invertebrates. Results may be enhanced by integrating mesocosms and field components for validation, addressing functional, behavioural and chronic endpoints (e.g., decomposition rates, avoidance, reproduction) and trophic interactions, screening and monitoring tools that are suitable at plot and field scales, and exploring the usefulness of modelling approaches in a variety of ecosystems. Greater ecological diversity that includes different functional groups and interactions between co-existing test species should provide results that offer a more robust ecological representation.

Expert opinion also concluded that various existing soil and aquatic biology and ecotoxicology indicators and methods may be suitable for evaluating soil-biochar-biota interactions, without requiring major adjustments or optimisation, besides possibly the recommended soil moisture content in standard testing (Busch *et al.* 2012). It is also likely that methods relying on the use of biomarkers (e.g., DNA, RNA, PLFAs; proteomics) may require optimisation. Efficient biomarker extraction may be compromised by adsorption to biochar surfaces, whereas interpretation of results may be biased due to confounding factors linked to biochar heterogeneity.

### 2.2. Soil organic matter and greenhouse gases

The highest RPIs were assigned to biochar interactions with soil organic matter (SOM) stability (priming), N<sub>2</sub>O and CO<sub>2</sub> fluxes and biochar C stability (Fig. 2), followed by CH<sub>4</sub> fluxes and cycling, and the nitrogen balance (e.g., plant N uptake, remaining soil N, and nitrate leaching losses). Plant-related indicators had the lowest RPI of this current analysis.

The decay of organic matter in soil is a complex process and is usually investigated indirectly by tracing one or more fluxes from the SOM pool over time (Kuz'yakov *et al.* 2014). For the decomposition of carbon-rich substrates, such as biochar materials, the prevalent mechanism is mineralisation to CO<sub>2</sub>, which has been emphasized as one important indicator. Assessing CO<sub>2</sub> fluxes over time is therefore considered to be a suitable experimental approach to quantify the degradation dynamics of biochars

added to soil, although some analytical difficulties remain to be overcome (Sagrilo *et al.* 2015). Biochars may affect the stability of biogenic soil organic matter: inhibit, have no effect or promote SOM degradation (Zimmerman *et al.* 2011; Zavalloni *et al.* 2011; Rittl *et al.* 2015; Ventura *et al.*

2015). Such priming effects influence the supply of nutrients (see 3.4). The effect of chars produced via hydrothermal carbonization (HTC) is less well investigated for the time being, but most studies suggest relatively low stability of hydrochars compared to pyrolysis biochars (Steinbeiss

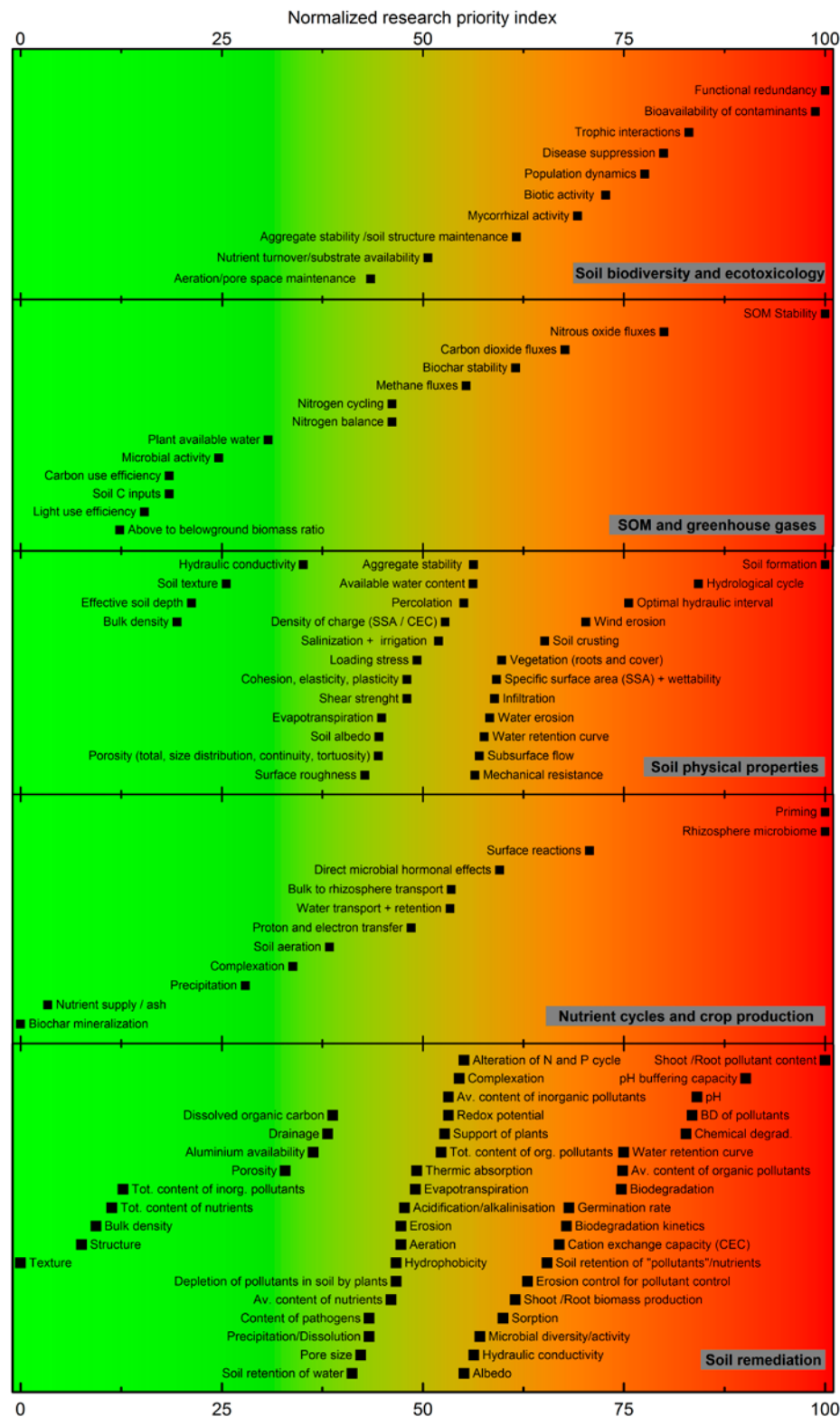


Fig. 2. Normalized research priority of the indicators as identified by the thematic groups. Higher values of RPI (and red background) refer to higher priorities. Abbreviations: Av. = Available; BD = Biodegradability; Org. = organic; Tot. = Total

*et al.* 2009; Qayyum *et al.* 2012; Dicke *et al.* 2014; Lanza *et al.* 2015; Schimmelpfennig *et al.* 2014; Busch, Glaser 2015), even when the hydrochar was carbonized at high-temperature (300 °C) plus high-pressure (30 bar) conditions (Schimmelpfennig 2015).

The stability of char materials in soil is a crucial indicator. Char stability can be estimated by the decay half-life or mean residence time. The use of biochar as a tool for C sequestration requires half-life values of at least several decades (Lanza *et al.* 2015), and a considerable body of evidence exists that biochar has such a half-life or more (Qayyum *et al.* 2012; Singh *et al.* 2012; Kuzyakov *et al.* 2014). However, these studies all rely on an extrapolation from investigations of limited duration of a few months to 8.5 years. Lehmann *et al.* (2015) undertook a comprehensive exploration of the available literature, normalized to an incubation temperature of 10 °C, the mean annual soil temperature of temperate regions. They found that the persistence of most biochars, particularly those with an H/C<sub>org</sub> ratio <0.4 is in the centennial range and thus sufficient for net C removal into a slower-cycling soil C pool. The stability of char in soil determines the length of time that any positive effects on soil quality and the environment may manifest.

Observations of the effects of biochar application to soil on N<sub>2</sub>O fluxes is not consistent but a reduction of N<sub>2</sub>O emissions is the most predominant result, particularly when the molar H/C<sub>org</sub> ratio is low (Kammann *et al.* 2012; Malghani *et al.* 2013; Cayuela *et al.* 2013, 2014, 2015; van Zwieten *et al.* 2015). Also CH<sub>4</sub> emissions from flooded soils can be reduced by improving CH<sub>4</sub> oxidation in the root rhizosphere, *i.e.* a biofilter process for anaerobically produced CH<sub>4</sub> before it leaves the sediment via aerenchyma (Jeffery *et al.* 2016.).

Different processes involved in the N<sub>2</sub>O production such as denitrification and nitrification are influenced by soil water status, bioavailable C content, pH, N availability and oxygen content. Limited understanding of the interactions controlling these processes limits the ability to predict biochar effect on N<sub>2</sub>O emissions (*e.g.* Sánchez-García *et al.* 2015). Meta-analysis predicts that woody biochars applied at rates above 1% by weight to a soil mixture have the potential to significantly reduce N<sub>2</sub>O emissions, particularly in non-flooded soils (Cayuela *et al.* 2014; van Zwieten *et al.* 2015). However, the mechanisms are still not well understood and range from pH effects (*e.g.* liming; Obia *et al.* 2015; Hüppi *et al.* 2015), to changes in soil N transformations (Nelissen *et al.* 2012), to shifts in the ratio of N<sub>2</sub>O/N<sub>2</sub> end products of denitrification (Cayuela *et al.* 2013). The hypothesis explaining the last mechanism assumes changes in denitrifier gene expression (Harter *et al.* 2014) and nitrate capture (Kammann *et al.* 2015). Besides mechanistic understanding, there is a definite lack in field experiments quantifying N<sub>2</sub>O emissions as compared to laboratory studies (Zhang *et al.* 2016). Long-term

field experiments considering the impacts of additions of organic and inorganic fertilizers on N<sub>2</sub>O emissions are needed for different agro-ecosystems in order to obtain a deeper insight into the mechanisms and potential utility of biochar additions for controlling N<sub>2</sub>O emissions.

### 2.3. Soil physical properties

The soil physical indicators requiring the most research to support policy and decision making were identified as soil formation, hydrological cycle, optimal hydraulic interval and wind erosion (Fig. 2). For most indicators, it was concluded that existing methods for measuring soil physical properties (*e.g.*, bulk density, soil moisture characteristics, mechanical resistance and shear strength) can still be used when biochar is present. However, this supposition should be validated for a variety of biochar types and addition rates. A possible concern is that the effects of biochar on soil indicators change as biochar ages, and an investigation of this would require longer-term field studies and artificial ageing studies in the laboratory (see also section 3.6). It was concluded that the effects of biochar ageing should be studied especially regarding the following soil properties:

- a) Particle size distribution;
- b) Water retention characteristics and plant-available water content of the soils;
- c) Cation exchange capacity (CEC);
- d) Porosity (total, pore size distribution and pore continuity);
- e) Saturated and unsaturated hydraulic conductivity;
- f) Least limiting interval range (difference between the minimum moisture content required to allow tillage and the moisture content at field capacity);
- g) Organic coating and biochar-mineral complex formation and its impact on nitrate capture by biochar particles.

Longer term soil formation could be affected by biochar soil amendment via changes in accumulation, transformation, and translocation of soil components leading to modified soil morphology and productivity (Spokas *et al.* 2012). Future work should include long-term field experimentation, exploration of soils around historic kiln sites (*e.g.* Borchard *et al.* 2014; Heitkötter, Marschner 2015) and artificial ageing methodologies that integrate physical, chemical and biological soil processes, as well as soil management.

The multi-faceted nature of the hydrological cycle makes evaluation of the effects of biochar on the hydrological cycle difficult (Kammann, Graber 2015; Masiello *et al.* 2015). The effect of biochar on soil hydrology depends on the feedstock and amount of biochar added, biochar quality including particle size distribution, pore size distribution, reactive surface area, and hydrophobic compounds on the biochar surface and soil type including soil water repellence, soil aggregation, soil bulk



density, and soil texture. Soil water content in soil-biochar mixtures is not always enhanced with the effects being dependent on application rate and usually with more pronounced increases in the macroporosity than the microporosity of soils (Tammeorg *et al.* 2014a, 2014b; Jeffery *et al.* 2015b; Masiello *et al.* 2015). The suggested use of biochar as a component of horticultural growing media (Méndez *et al.* 2015; Vaughn *et al.* 2015; Kern *et al.*, this issue) makes it necessary to differentiate between soils and growing media for calculating the influence of biochar on retention of plant-available water. In the latter case, it is advisable to determine biochar water retention between  $-1$  and  $-10$  kPa, differentiating between easily available water (from  $-1$  to  $-5$  kPa) and water buffering capacity ( $-5$  to  $-10$  kPa).

Regarding wind erosion, biochar with some particle sizes can become easily airborne if a moisture content of at least 15% is not maintained (Silva *et al.* 2015). A key requirement is the quantification of biochar separately from other forms of carbon in soil. Related to this are requirements to detect the movement of biochar in the soil profile and the transport of biochar particles by water and wind erosion. Recommended methods for distinguishing biochar from other carbon in soil include stable isotope technology in combination with isotopically-labelled biochar. If labelled biochar was not used, possible methods include chemical extraction methods of black carbon, e.g. by using benzene polycarboxylic acids as markers for biochar (Glaser *et al.* 1998; Brodowski *et al.* 2005). Similarly, indirect methods such as mid-infrared and near-infrared spectroscopy and multivariate data analysis (Bornemann *et al.* 2008; Allen, Laird 2013) can also be used, but still need to be standardized.

## 2.4. Nutrient cycles and crop production

We identified priming of SOM, rhizosphere microbiome, surface reactions and direct microbial hormonal effects as being the most critical indicators for future research (Fig. 2). Rhizosphere microbiome and biodiversity were considered within the context of “How does the impact of biochar on biodiversity affect nutrient cycling and the way it can be evaluated.” Potential suggested indicators were microbial functioning assessed by molecular markers, cultivation techniques and enzymatic methods.

As SOM is the most important source of nutrients such as N and P (Blagodatskaya, Kuzyakov 2008), effects of biochar on SOM turnover (priming effects) are likely to affect the availability of soil nutrients derived from SOM mineralization, e.g., by speeding up nitrification (Sánchez-García *et al.* 2015). Therefore priming effects were considered highly important. However, both negative and positive effects on SOM stability have been reported in the literature (Maestrini *et al.* 2015) indicating that biochar can both

inhibit, or promote, or have no effect on SOM degradation. The effects depend both on the length of the incubation study (Lehmann *et al.* 2015), and the presence or absence of plants (Weng *et al.* 2015; see also section 3.2).

Some observations indicate that biochar reduces nitrate leaching from soil (Ventura *et al.* 2012; Laird, Rogovska 2015; Haider *et al.* 2016) but this does not translate directly into improved crop growth. Biochar increased N utilization efficiency, but reduced N accumulation in plants, probably because some mineral N was captured by biochar (Zheng *et al.* 2013; Haider *et al.* 2015). If captured N during a pre-loading (or post-production treatment) process by biochar is easily released to plants, the loaded biochar may increase plant growth since it then acts as a slow-release fertilizer (Kammann *et al.* 2015) and indeed biochar has been investigated as a support material for slow release mineral fertilizer (González *et al.* 2015).

Biochar has been shown to increase the activity of different soil enzymes (Bailey *et al.* 2011; Ventura *et al.* 2014). In general, also microbial biomass increases after biochar application (Liu *et al.* 2016) suggesting that biochar may promote nutrient cycling in soil. For example, biochar may promote P mobilisation by stimulation of soil microbial activity, although the response is strongly dependent on soil type (Deb *et al.* 2016). Understanding the effect of biochar on the soil biological community (and on the rhizosphere microbiome in particular) is therefore critical, for the development of on-farm soil management and conservation practices to improve soil properties, agricultural productivity and environmental performance. However, the effects of biochar on the rhizosphere microbiome remain uncertain meaning that there is an important gap between the current and required LOSU's.

A better knowledge of plant-biochar-microbial interactions may eventually lead to applications of biochar as a carrier of beneficial microorganisms and thus reduce the use of current carriers like peat, vermiculite or perlite (Hale *et al.* 2015). Recently, it has been shown using P isotopes that mycorrhizae can actively mine biochar pores for (loaded) phosphorus (Hammer *et al.* 2014). Other trials have focused on selecting the most efficient strains of microorganisms to utilize nutrients contained in biochar (Postma *et al.* 2010). Therefore there is potential to develop new microbiologically enriched biochar preparations.

Biochar can itself be a source of nutrients such as P (Jin *et al.* 2016). However, the nutrient content of biochar is generally low, with the exception of ash-rich biochars produced from nutrient-rich feedstocks (Glaser *et al.* 2002). Moreover, nutrients such as nitrogen (N) are mainly bound to biochar covalently and so not immediately available for plant uptake because their mineralisation rate will be slow. For these reasons, direct nutrient supply via biochar mineralization was considered less important in comparison to other indirect processes.



Biochar may affect soil nutrients by reducing leaching losses from soil (Biederman, Harpole 2013; Ventura *et al.* 2012) and its surface reactivity is likely to be the most important factor influencing this process. However, biochar increases soil water retention and this could therefore reduce nutrient leaching by reducing water movement through soil (Laird *et al.* 2010). The effect of biochar on water transport and retention needs to be investigated to determine its effect on nutrient losses.

## 2.5. Soil remediation

We identified the following indicators as the most important research priorities: content of pollutants in plants (both in shoots and roots), soil pH value and buffering capacity, and dynamics of organic pollutants (Fig. 2). It is critical to evaluate whether biochars can increase or decrease the uptake of organic and inorganic pollutants by plants, both in the context of clean crop production and phytoremediation. Translocation of metals from roots to shoots has been shown to vary in the presence of biochar in some cases (Rees *et al.* 2015), however, investigations of the precise localization of pollutants in the different plant tissues have seldom been made. For some volatile organic pollutants, release of pollutants accumulated by plants may occur via evapotranspiration or outward diffusion from above-ground biomass (Sorek *et al.* 2007). Biochar might affect this process through adsorbing the compounds in the root zone, thereby reducing phytoavailability, or by affecting plant metabolic processes, but few related investigations have been made so far. The effect of biochar on root development, e.g. root surface, has also rarely been described, despite its importance regarding pollutant uptake (Hammer *et al.* 2014; Graber *et al.* 2015; Rees *et al.* 2016). All these plant-related research topics should be targeted in the near future to improve understanding of biochar's effect in vegetated soils (Kammann, Graber 2015). Besides plants, other organisms should be included in experiments involving biochar in soil remediation. The potential synergetic or antagonist effects of biochar and *Arbuscular mycorrhizae* on the bioavailability of potentially toxic elements should be elucidated in controlled and field experiments. The effects of earthworms on urban soils and their synergy with biochar regarding soils remediation are also poorly known (Beesley, Dickinson 2011; Gomez-Eyles *et al.* 2011).

Although pH is one of the key parameters that control the chemical and biological transformations of pollutants and their mobility in soil, soil pH evolution following biochar amendments has not always been properly monitored and remains poorly predicted. One of the main associated issues is to understand how long biochar will maintain a sufficiently high soil pH in metal-contaminated soils to limit the mobility of metals. Soil pH buffering

capacity should be measured after biochar addition over a long period of time (more than 2 years for field experiments) to assess the influence of biochar ageing. Complementary to pH, measurement of redox potential could provide valuable information about microbial activity in remediation processes and in any case soil redox potential can be directly affected by biochar (Joseph *et al.* 2015). In this case, amendments of biochar should be tested under dynamic redox conditions, for example in contaminated floodplain soils.

Regarding organic pollutants, biochar may sorb organics at its surface and limit not only their mobility but also their biodegradability (Mumme *et al.* 2014). The monitoring of both biotic and abiotic degradation of organic pollutants is often difficult and depends on the mass transfer limitations in soil-biochar media (Gul *et al.* 2015). Information about the degree of abiotic transformations (e.g. sorption, complexation, precipitation) could be produced by using sterile media (soil and biochar).

Besides heavy metals, metalloids and organic pollutants, the use of biochar for the immobilization of radionuclides may be suggested. Biochar-assisted soil remediation sometimes implies finding a compromise between the immobilization of certain pollutants and the mobilization of others (e.g., Cd, Zn or Pb versus As). The application of biochars designed to immobilize pollutants may also cause the undesirable immobilization or inactivation of other compounds, such as fertilizers or pesticides (Beesley *et al.* 2011). In the same way, the possible retention/protection of specific microorganisms, including pathogens, within biochar particles should be addressed when implementing biodegradation strategies. Finally, the influence of biochar ageing deserves particular attention when addressing soil remediation. Aged biochars often have greater concentrations of carboxylic functional groups on their surface, which can serve as additional binding sites for metal ions (Qian, Chen 2014; Wiedner *et al.* 2015) but more research is needed to understand ageing effects on metal interactions with biochar (Puga *et al.* 2016). Ageing also increases sorption of some herbicides whereas others are better adsorbed on fresh biochar (Trigo *et al.* 2014). One way to investigate biochar's ageing effects is to use composting which is suggested to accelerate the ageing of biochar (Kammann *et al.* 2015; Wiedner *et al.* 2015).

## 2.6. Integrated discussion

### 2.6.1. Thematic group perspectives

Figure 2 shows soil properties or processes assigned to research priorities anticipated to close the LOSU gap across the themes. Not surprisingly, given that the discussions were focused on particular themes, different thematic groups assigned individual properties and processes differently. One thematic group may have perceived that

understanding a specific property or process is a high priority when considering their theme, while another may have concluded that the same property is less important to their theme, even if both recognise the overall scientific understanding of the property or process similarly. For example, soil structure has a green RPI score of 8 for “soil remediation” but for “soil biodiversity and ecotoxicology” it is amber (RPI 62), as are aggregate stability, mechanical resistance and soil crusting under “soil physical properties”. On the other hand, a maximum RPI score of 100 was assigned to SOM stability/priming by both the groups considering the themes “SOM and greenhouse gases” and “nutrient cycles and crop production”.

### 2.6.2. Cross-cutting themes

Five thematic group sessions combined with repeated plenary sessions, highlighted four key issues that were identified across the different thematic groups.

Firstly, one of the most common recommendations to close the LOSU gap was to upscale experiments and move from short-term, laboratory-controlled conditions to long-term, field experiments (ideally even at catchment scale). Long-term experiments are particularly important regarding the understanding of biochar’s ageing, even if artificially accelerated ageing of biochar may provide complementary answers (Zhang *et al.* 2016). Investigations across a range of soil types and climatic conditions are needed, particularly as there may be a current bias towards high quality temperate soils, while the benefits of biochar application to soil quality may be more useful in nutrient-poor, degraded or acidic soils (Glaser *et al.* 2002; Jeffery *et al.* 2011).

Secondly, there are analytical methods developed for soils that appear to be valid when biochar is present, while others require adjustment to improve measurement accuracy. Introducing biochar into soil poses additional methodological challenges, such as its distinct quantification from SOM or its separation from soil particles for characterization. Biochar should be characterized prior to its addition to soil using established methodologies, e.g. mechanical resistance, density, porosity, chemical composition and other parameters listed in the various biochar characterization schemes (EBC 2012; British Biochar Foundation 2013; International Biochar Initiative 2015; Bachmann *et al.* 2016), as well as new ones, according to the research need. Analytical biochar characterization should be complemented by effect-based approaches in soils that are reflective of possible risks, as has been promoted throughout the COST Action TD1107 Representative sampling of large quantities of biochar represents an additional challenge to achieving reproducible results. The representative sampling practice suggested by Bucheli *et al.* (2014) is time-consuming but necessary when large

variability exists among the measured properties of sub-samples. A further recommendation is for biochar producers to install incremental cross-stream sampling devices to provide representative sampling (Bucheli *et al.* 2014).

Thirdly, the dust from biochar may be of concern in relation to both human exposure and climate feedback. The exposure of a population (mainly rural) in areas where biochar is to be applied, such as through inhalation due to wind erosion, should be minimized by moistening the biochar before application and mixing it with the topsoil as quickly as possible (Silva *et al.* 2015). However, moistening during application does not eliminate the risk of exposure during subsequent years due to wind erosion. Probably, the hazard represented by wind erosion depends mainly on the biochar application rate and its content of pollutants, but this issue should be addressed more thoroughly. Furthermore, black carbon aerosols may reduce the climate change mitigation potential of biochar (Genesio *et al.* 2016) and the LOSU regarding how and to what extent this may occur is close to zero.

Finally, an aspect of the LOSU concept is the representability of environmental and management factors. This is an area where science and policy are not aligned. The scientific model rewards innovative research, while policy requires trials to demonstrate reliability and studies that extend the experimental knowledge base to all the environmental and management factors relevant to policy development. Edwards (2016) makes a strong case that industry needs to get more involved, by forming partnerships with scientists, and possibly governments. Edwards (2016) argues that a research charity funded by industry, and possibly governments and other charities, would “create a system that rewards science that is both cutting-edge and reproducible”, as long as it is based on sound principles of data sharing, public quality criteria, independent oversight, public ownership of outputs, etc. An academia-industry partnership on biochar in this way may be a useful catalyst to bridge the LOSU gap.

## Conclusions

In recent years, research activity on the use of biochars in soils has been increasing and this trend is likely to continue over the next decade due to the numerous potential benefits and risks associated with the use of biochars. An optimal allocation of research resources to resolve outstanding issues is essential for the application of these materials to soils to be progressed efficiently. This requires clarity about research priorities.

Based on the specific expertise and interdisciplinary representation of the participants in the Aveiro meeting, we propose that the most critical topics for future research regarding biochar soil application are as follows:

- Functional redundancy of soil biota, and bioavailability of contaminants present in biochar.

- SOM stability and N<sub>2</sub>O emission reduction.
- Soil formation, the hydrological cycle and soil water supply to plants.
- Priming of SOM and modifications of the rhizosphere microbiome.
- Plant uptake of pollutants and soil pH buffering capacity.

However, gaps remain in relation to how biochar affects most soil properties and processes. We hope that the identification and prioritization of gaps in our current LOSU, along with the identified key issues, will be a stepping stone on the path to reaching the required LOSU for the development of a sustainable biochar application system by scientists for practical users of biochar.

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